



# Black-legged kittiwakes in Glacier Bay National Park and Preserve

*A review of existing data and recommendations for long-term population monitoring*

Natural Resource Technical Report NPS/SEAN/NRTR—2013/672





**ON THIS PAGE**

Black-legged kittiwake, Tarr Inlet, Glacier Bay National Park and Preserve

Photograph by: Christopher J. Sergeant, Southeast Alaska Inventory and Monitoring Network

**ON THE COVER**

Nesting black-legged kittiwakes, Glacier Bay National Park and Preserve

Photograph by: Aleya Brinkman, University of Alaska Fairbanks

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# **Black-legged kittiwakes in Glacier Bay National Park and Preserve**

## *A review of existing data and recommendations for long-term population monitoring*

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## Executive Summary

This project initiated the process of updating an existing long-term monitoring protocol for black-legged kittiwakes (*Rissa tridactyla*, henceforth kittiwakes) as part of a larger monitoring effort, the marine predators vital sign program, in Glacier Bay National Park and Preserve (GLBA). Production of the original monitoring protocol for kittiwakes was spearheaded by Hooge et al. (1998) during the 1990s. The purpose of this earlier work was to develop an efficient survey methodology to annually census/monitor kittiwake populations in GLBA at their breeding colonies. Hooge et al. (1998) compared the accuracy of direct visual counts versus counts from photographs. The authors used a repeat survey design and examined the minimum number of replicate surveys necessary for various levels of precision. Additionally, they presented process-based information on the equipment, frames, slide counting, and data management components of the monitoring.

In 2011, the Southeast Alaska Inventory and Monitoring Network (SEAN) established a task order under a Cooperative Ecosystem Studies Unit (CESU) agreement with the University of Alaska-Fairbanks (Agreement Number J9910324812, Modification 3). This report is the final product of that task order, the objective of which was to initiate the protocol development process for a long-term kittiwake monitoring protocol by reviewing the existing protocol with respect to advances in sampling techniques, photographic monitoring, and population modeling. We conducted a literature review, an assessment of historical data, preliminary data analysis, and field sampling. We provide a summary of our findings and recommendations for future steps.

The literature review spans two categories: ecology of kittiwakes and photographic sampling methodologies for colonial nesting birds. In the ecology section, we provide the justification for using kittiwakes as indicators of the marine environment as well as several views on the status of Pacific populations. We identify different biological units of interest that may be appropriate for specific questions. In the methodology section, we consider numerous applications of photographic sampling to a variety of bird species and elucidate elements of the photographic process that can induce substantial sampling variation or bias. We provide a section on monitoring protocol considerations identifying several modeling frameworks that may be appropriate for sampling and analyzing survey data on kittiwakes in GLBA, address multiple statistical considerations, and detail requirements for development of a long-term monitoring protocol.

We have included two appendices which are referenced throughout the document. Appendix A refers to our assessment of the historical data, and Appendix B presents results from our 2011 sampling. The historical data presented in Appendix A comprises approximately 1,300 35mm color slides taken at the Margerie Glacier, Kashoto Bay, and Lone Island colonies from 1993 to 1995. We scanned all of these slides at high resolution where they are archived at University of Alaska Fairbanks and with SEAN at the Glacier Bay Field Station in Juneau. As of this report date, we have processed two out of thirteen plots from the Margerie Glacier colony; this processing entails digital photo-stitching, establishing equivalent area subplots, and performing counts of nests. Our analysis indicates that there is large annual variation in abundance and occupancy of nest sites, which has implications for monitoring protocols. Some of this variation can be attributed to the photographic process, which accounts for differences in light conditions, plot coverage, framing photos in high nest density areas, focus, and photographer perspective.



Individually, none of these variables rose above another in explaining the variation seen in counts, but they are collectively substantial when considering approaches to long-term monitoring. This analysis suggests that photographic methods (e.g., framing, exposure settings) will influence the overall quality of images, thereby impacting counts generated.

Appendix B details 2011 field sampling efforts. In June 2011, we conducted three repeat photographic counts at three separate colonies in GLBA toward the end of the kittiwake incubation phase. We varied parameters of the photographic process such as camera aperture and focal length to better understand sampling variation as indicated by our preliminary data analysis. Additionally, we confirmed the presence of breeding kittiwakes at several colonies that had been declared declining in 1995.

To continue development of a robust, long-term monitoring protocol for kittiwakes, we suggest the next steps: refine potential sampling and analytical methods by performing further analyses of the historical dataset with comparison to photos taken in 2011, and develop a standard operating procedure to enable replication of photo acquisition and processing.



# I. Literature Review

## A. Ecology

Black-legged kittiwakes (*Rissa tridactyla*; kittiwakes) are a small, abundant gull (family *Laridae*) of the North Pacific and North Atlantic. In general, they are long-lived and do not start breeding until 4–5 years old (Suryan and Irons 2001). Over the years, much research effort has been directed toward studying these birds due to their wide range, colonial nature and fidelity, and ease of viewing and identification. It is thought that kittiwakes are highly sensitive to marine conditions and can provide insight into oceanographic variability (Hatch et al. 1993a).

Kittiwakes feed at surface waters near or moderately near to breeding colonies. They procure their prey by dipping, surface seizing, and plunge diving. They target sandlance (*Ammodytes* spp.), capelin (*Mallotus villosus*), juvenile cod and pollock (*Gadidae*), and lanternfishes (*Myctophidae*; Hatch et al. 1993a). Murphy et al. (1991) suggest that it is actually availability of these forage fishes that determines reproductive success. Prey availability affects kittiwake diets, foraging trip duration and provisioning, and growth and survival of nestlings (Suryan et al. 2002). Examining whether prey productivity similarly influenced other dimensions of the ecosystem, Hatch et al. (1993a) investigated the reproductive success of common and thick-billed murres (*Uria* spp.), horned and tufted puffins (*Fratercula* spp.), and pelagic and red-faced cormorants (*Phalacrocorax* spp.). They found that none of these birds exhibited the high levels of variation seen in annual productivity of kittiwakes; therefore, these other species do not appear to be affected to the same extent by the conditions that impact kittiwakes. In addition, kittiwakes have a high level of site fidelity to a colony (Hatch et al. 1993b) even if there is poor reproductive success (Suryan and Irons 2001). Suryan and Irons (2001) speculate that this may be because they also have high mate fidelity and foraging patterns may be learned and colony-specific such that emigration incurs feeding and breeding costs. Consequently, kittiwake ecology may more closely reflect ambient marine environmental conditions than other candidates for long-term monitoring, particularly at the colony scale.

Suryan and Irons (2001) discuss how decadal shifts in oceanographic conditions affect kittiwake ecology. Due to physical oceanographic processes related to the Pacific Decadal Oscillation, the timing of primary production shifts and results in annual changes to the accessibility of phytoplankton biomass (Suryan et al. 2006). Phytoplankton is necessary for the over-winter survival of young of the year (YOY) fishes and it is one-year old fishes that are most important to kittiwakes. These prey species are available at critical times, such as the first few weeks of nestling growth in the beginning of July, and are more nutritious than YOY fishes. During the 1970s colder than average temperatures (at sea surface, 150 m and 250 m depth) were recorded, whereas warmer than average conditions were observed in the 1980s. It is these temperature swings, driving reorganization in the community structure of fishes, which alter the availability of one-year old fish to kittiwakes (Suryan et al. 2001). For example, during the 1980s in Prince William Sound, forage stock such as sandlance, pandalid shrimp (*Pandalus* spp.), and capelin were depleted by predatory fishes such as cod and pollock (Suryan and Irons 2001). This shift was thought to affect marine mammals and birds resulting in widespread declines across multiple species. Suryan and Irons (2001) suggest that decadal shifts in oceanographic conditions generate cycles in kittiwake productivity.

Hatch et al. (1993a) used breeding colony counts to estimate a population of 2.6 million Pacific black-legged kittiwakes distributed across the Bering Sea, Gulf of Alaska, and Alexander

Archipelago. While some breeding colonies are exceedingly numerous (>10,000 birds), the largest in Glacier Bay National Park and Preserve (GLBA) is comprised of approximately 4,000 birds and 3,000 nests (Lentfer 1992). In Pacific populations, black-legged kittiwakes have exhibited poor reproductive success since the 1970s (Hatch et al. 1993a). The mean annual productivity of kittiwakes in 162 colony years up to 1989 was 0.31 chicks/pair. Though overall declines in the population have not been widely documented, populations cannot be sustained with these reported rates of productivity without commensurate increases in survival or immigration. Hatch et al. (1993a) surmise that population decline has not been as apparent because high levels of adult survival are compensating for successive breeding failures. Hatch et al. (1993b) found that annual adult survival was 0.92, a rate which predicts an average reproductive life of 13 years.

For kittiwakes, productivity is serially influenced by “ability to achieve breeding condition” (Hatch et al. 1993a). Whether an adult reaches breeding condition is perhaps a factor of spring environments. The consequences of poor breeding condition can manifest as delayed breeding, smaller clutch sizes, fewer nest and breeding attempts, or lack of breeding altogether (Murphy et al. 1991). Following egg-laying, productivity is next influenced by success in hatching and raising chicks (Hatch et al. 1993a). Negative impacts at either breeding stage can result in colony breeding failure. Hatch et al. (1993a) defines colony breeding failure as a mean colony productivity of  $\leq 0.1$  chicks per nest with maximum observed rates of 1.92 chicks per nest. Typically, smaller colonies tend to have lower reproductive success but, in Alaska, fledging success (the number of chicks fledged per nest) is often characterized by frequent low and total failure (Suryan and Irons 2001).

While colony breeding failure is widespread across the Pacific Ocean, from the Bering Sea and Gulf of Alaska to Prince William Sound and Glacier Bay, high annual and interannual variability leads to difficulty in classifying the scale at which these colony breeding failures occur. For example, Murphy et al. (1991) documented multiple complete or mostly complete reproductive failures at colonies in the Bering Sea from 1979 to 1989. Yet in Prince William Sound, Suryan et al. (2006) documented high levels of reproduction during the 1980s, while the 1990s were a reduced period of reproductive output. Given these differences across large geographic areas, a conservative approach to identifying trends would be to assume that kittiwake colonies are primarily affected by ambient surroundings. However, uncertainty remains in determining which scales are most influential because mechanisms that affect reproductive success can be interactive or region- and site-specific (Suryan et al. 2006). This complexity results in difficulty characterizing the extent of colony failures and, ultimately, their cause.

There has been much investigation into identifying behavioral cues that predict the eventual colony reproductive outcome, with no clear conclusions. This could be a function of the difficulty in viewing nestlings and separating survival from hatching, fledging, and non-detection. Gill et al. (2002) echoes others’ findings that kittiwakes are highly sensitive to food availability during the breeding stage, but also demonstrates uncertainty in which breeding parameters are thought to be the most sensitive and, therefore, the best gauge of the food conditions. Potential parameters of interest could be nest relief (exchange of adults at nest for foraging), attendance to chicks, courtship, chick provisioning rates, and sibling aggression. Gill et al. (2002) found that the conventional measures hatching, fledging, and productivity best described success, but were not useful in predicting the reproductive outcome of the colony.

However, they did find that the time allocation of adults was correlated to prey availability. They speculated that when resources are limited, adults spent more time off the nest foraging, which decreased chick provisioning rates and impacted chick growth. This hypothesis is supported by Suryan et al. (2006) who found that the duration of feeding bouts during the incubation period is closely related to duration of feeding bouts during the chick-rearing. They suggest that adult kittiwakes may have some clue during incubation that reduced prey availability will affect their ability to provide for their chicks, and adults may differentially commit their effort to chick-rearing. This suggestion is supported by the observation that kittiwakes have more frequent nest relief in periods of greater food availability, exchanging nest duties at 2–3 times higher rates (Gill et al. 2002), with greater incidence of provisioning during chick rearing. This observation contrasts with what Coulson and Johnson (1993) found in North Shields, England where they investigated nest attendance and concluded that within colonies there was high variability in nest attendance in a single year. They speculated that this reflected individual differences between birds rather than a relationship to abundance of prey. The authors state that “caution is required before it is concluded that parental absence is an indicator of food storage.” Neuman et al. (1998) evaluated whether courtship feeding could be used to predict prey availability. They hypothesized that male-supplied food boluses play an important role in provisioning females to enhance breeding condition. However, they did not find direct evidence that courtship feeding influenced breeding success. They caution that their study was conducted during a year of high reproductive success and speculate that courtship feeding may be integral in years when food is limited. While multiple papers have investigated a variety of behaviors, general conclusions are uncertain. No single cue has been identified that predicts reproductive outcome and can serve as a guide in allocating monitoring resources to estimate colony productivity.

## **B. Photographic Sampling Techniques for Birds**

Traditional approaches to sampling avian populations of waterbirds and seabirds have involved extensive efforts to make direct counts at breeding colonies (Hutchinson 1980). However, there has been high variability associated with this process due to factors such as survey timing, detectability, double counting, observer bias, and disturbance (Frederick et al. 2003, Robertson et al. 2008). These difficulties underscore the need for more precise and cost-effective methods to estimate abundance.

The use of photography in monitoring avian populations to mitigate these challenges is becoming widespread. Species that breed or stage colonially, such as waterbirds and seabirds, lend themselves well to photography. Aerial-, ground-, and boat-based photography have been used to survey cormorants, auklets (*Aethia* spp.), murre, albatrosses (*Thalassarche* spp.), cranes (*Grus* spp.), ducks (*Aythya* spp.), geese (*Branta* spp. and *Chen* spp.), gulls (*Larus* spp.), kittiwakes, herons (*Ardea* spp.), and penguins (*Eudyptes* spp.). Photographs may allow researchers to sample populations with less variation than other techniques while creating a permanent historical record to be resampled at any future time for diverse reasons (Nettleship 1978).

There are several points to consider when using photography to survey birds. First, species life history and logistics will dictate whether photography is appropriate for a population or time period. Second, if photography can be used, variation introduced at multiple stages of this process needs to be considered. The timing, shooting, and sampling of photographs each have components that can affect counts. By anticipating circumstantial effects and standardizing

photographic survey methods, researchers can reduce the impact of known or potential sources of bias. Ultimately, photographs of the target population will capture the true number of birds present and can be used to account for detection and estimate abundance.

The accuracy of photographic counts is a function of biological factors such as survey timing and behavioral patterns of the target species. For species whose nests are easily identifiable, such as common murre (*Uria aalge*) and Brandt's cormorants (*Phalacrocorax penicillatus*) on the Oregon coast sea stacks or black-legged kittiwakes on vertical cliffs along the Alaska coast, variation in counts can first be attributed to species ecology. Surveys should be conducted during peak conditions for presence (Johnson and Krohn 2001, Seavey and Reynolds 2009), which could be either during the later stages of incubation when breeders exhibit high nest fidelity, or after all seasonal migrants have returned to staging areas. Diurnal behavioral patterns can result in different numbers of birds present at the colony at different times. For kittiwakes, the best time to survey is between 0900 to 1600 hrs (Hatch and Hatch 1988, Wanless and Harris 1992, Coulson and Johnson 1993). Estimating the probability of presence for specific daytimes is not usually feasible; therefore, efforts should be made to target maximum numbers.

Using photography to evaluate reproductive success has not been effective (Hooge et al. 1998). Hatching success is hard to determine because direct observation of chicks may be difficult due to presence of brooding parents, cryptic coloration of chicks, and angle of observation relative to the nest cup itself. In addition, distinguishing between non-detection and chick death can be difficult. Though the first two weeks of chick survival are the most critical (Hatch et al. 1993b), reproductive success is usually evaluated when chicks are older and more visible, but have not yet fledged. Consequently, photo censuses are generally of breeding adults and annual changes represent recruitment, dispersal, or habitat quality.

Secondly, aspects of the physical environment may affect photographic counts such as topography and vegetation. Aerial photography is used to provide perspective for otherwise inaccessible or unnavigable landscapes and is particularly suitable for complex coasts, large study areas, and for species that are disturbed by the proximity of ground- or boat-based researchers. Individual snapshots or continuous video are most commonly conducted by fixed-wing aircraft with the occasional use of helicopters. Cameras are mounted to ensure consistent perspective and can be linked to a GPS to capture spatial information with each frame (Wilson 2010). Aerial photography is usually corroborated by ground-truthing to provide some estimate of the proportion of the population that is missed by the photography, which, in turn, is used to correct counts obtained from photographs. Another use of ground-truthing aerial photos is to calculate the footprint of a photo. By flying at exact elevations and angles to the ground, the area covered by the photo can be estimated and a density calculated (Anthony et al. 1995).

Boat-based photography is best-suited to colonies which occupy vertical cliff faces such as kittiwakes. However, the researcher will need to consider the angle which maximizes the probability of detection, particular as terrain grades away from the observer (Kirkwood et al. 2007). Perpendicular angles for each photo are best to increase sighting of birds and nests (Hooge et al. 1998). Photographic angle is especially important for cryptic species that can be concealed by vegetation or crevices where detection is frequently less than one. Greater black-backed gulls (*Larus marinus*), herring gulls (*Larus argentatus*), and glaucous-winged gulls (*Larus glaucescens*) nest in or near vegetation, which can change markedly over the course of

the breeding season (Johnson and Krohn 2001). Rockhopper penguins (*Eudyptes chrysocome*) in the south polar sea nest in extremely rocky terrain where crevices obscure birds on nest (Kirkwood et al. 2007).

The photographic process itself can also introduce variability. Blurry or distant photos and incomplete colony coverage may affect counts by making nests too small or missing them entirely, underestimating the population. Weather conditions are highly influential; sun glare can shadow or overexpose objects while precipitation can inhibit the use of sensitive camera equipment. Haramis et al. (1985) found that overcast days are best for shooting canvasbacks (*Aythya valisineria*) in Chesapeake Bay because water reflection was minimal.

Once photographs have been taken and are ready to be processed, variation from the counting process needs to be considered. Direct counts and counts generated from sampling photos can vary with regard to individual observer. Observers may differ in their ability to identify species and estimate flock sizes (Frederick et al. 2003). Observers can also develop a familiarity with certain nest sites such that subsequent counts diverge from initial counts. The conventional approach to restricting variability between observers is to perform multiple counts with different observers. For censusing kittiwakes in GLBA, differences between observer counts less than 5% were averaged while differences greater than 5% were repeated until acceptable (Hooge et al. 1998). Averages between observers and multiple counts have been typical (Johnson and Krohn 2001, Robertson et al. 2008, Renner and Renner 2010) while some effort is being made to model and correct for individual observer differences (Gibbs et al. 1988, Anthony et al. 1995, Frederick et al. 2003).

Photo processing and counting can be labor intensive or limiting when large areas or high numbers of birds are involved (Gillmer et al. 1988). Prior to the advent of digital photography, photos were enlarged (or color slides projected) and covered with clear transparencies; birds or nests were then tallied with hand-held counters (Dolbeer et al. 1997). Wintering Chesapeake Bay canvasbacks were manually marked and sexed on photos by Haramis et al. (1985) to evaluate latitudinal differences in sex ratios. This method excluded several flocks of greater than 10,000 birds which were too numerous for full coverage and complete counts. Various attempts at automating this process have been made to reduce the high cost of manually counting. Tratham (2004) used the Image Processing Toolbox in MatLab to specify color thresholds and pixel configurations that indicated macaroni penguins (*Eudyptes chrysolophus*) at their breeding colonies in South Georgia. To identify white phase snow geese (*Chen caerulescens*) in the St. Lawrence River watershed, Bajzak and Piatt (1990) summed all pixels within the color threshold then divided by the average size (in pixels) of a snow goose. The resulting estimate was within 2.3% of ground-truthed counts. The automation process is not suitable for every species and system. Analysis of photos is usually based on color bands and animal shape therefore, species need to be distinct from their surroundings and maintain a consistent size.

As digital photography and image processing software become ubiquitous and storage capabilities increase more attempts will be made to census or sample colonial populations in this manner.



## II. Monitoring Program Considerations

The foremost priority to address when developing a monitoring protocol for any species is the purpose of the work and specific, measureable monitoring objectives (Yoccoz et al. 2001, Oakley et al. 2003). Investigators must specify the type of information needed, the nature of the data required to generate that information, and the nature of the sampling design and field effort required to generate those data in a statistically valid manner (Moynahan and Johnson 2008). For example, there are several broad potential objectives that arise with respect to black-legged kittiwakes in GLBA: determining population status and trends, evaluating reproductive success in terms of oceanographic conditions, and/or documenting colony movement in response to glacial succession. The formulation of a study design will be carefully linked to these long-term objectives. Other aspects to consider in developing a monitoring protocol are the various sources of variability and heterogeneity, development of field protocol (procuring the necessary and highest quality data, previously discussed in sampling techniques), analyzing and reporting the data, evaluating the effectiveness of the monitoring over time, and ultimately incorporating findings into a management or conservation plan.

Another critical step in initiating a monitoring program is to determine the most biologically significant unit of interest to monitoring a specific population. For kittiwakes, Hatch and Hatch (1988) discuss three main units: nest counts, daily bird counts, and counts of occupied sites. Nest counts are direct counts of the number of breeding birds. Nest counts typically do not vary much over the short-term, but must be well-timed in order to accurately reflect colony dynamics. Actual counts of birds give both the number of breeders and non-breeders, if non-breeders are present. Bird counts can be very sensitive to both daily and seasonal variation necessitating multiple counts in order to separate trends from sampling variation. For example, nonbreeding birds spent up to 39% of their time at colonies in the Semidi Islands, Gulf of Alaska, while breeding birds were present 50% of the time (Hatch and Hatch 1988). The occupied site is any site that has a single or pair regardless of nesting material. This parameter is not sensitive to failure of nests, failure to build a nest, or failure to lay eggs. For example, in 1979, Hatch and Hatch (1988) found that a large number of pairs failed, but still attended their nest sites regularly. Overall, they found that the number of nests built and the number of occupied sites were more variable on an annual basis than daily counts of birds. They suggest that for population monitoring daily counts of adult birds throughout the incubation stage are most useful for detecting annual changes in the number of birds. However, this recommendation brings us back to the lingering question: does a decrease in the number of nesters represent a population decline and will our biological unit capture this? Regardless of the definition, the final unit decided upon needs to reflect the monitoring objective closely.

While to some degree the monitoring objective will dictate the specific study design, modeling process, and selection of the most appropriate sampling unit, there are some general guidelines for sampling under different approaches. Occupancy modeling and repeat counts offer considerable promise in applications for long-term monitoring of species such as kittiwakes. An occupancy modeling approach will yield information on probability of presence, colony configuration, nest fidelity, and species persistence, based on detection-non-detection data. Traditional uses of occupancy modeling have been to determine species occurrence, range, and distribution (MacKenzie and Royle 2005), and occupancy surveys are also useful for species that are difficult to detect that may be rare or elusive (Thompson 2004, MacKenzie et al. 2002).



Occupancy modeling is useful in long-term monitoring programs as long as the unit of interest – occupancy – is in accordance with program objectives (MacKenzie and Royle 2005). Some attempts have been made to scale occupancy to abundance (Royle et al. 2005); however, when abundance is the parameter of interest, studies should be designed to estimate abundance directly.

The sampling units for studies of occupancy are dependent upon both species and objectives, which could include geographic or political areas, wetlands, habitat patches, or even nests. Repeat surveys are necessary to estimate occupancy and detection (Nichols et al. 2009). Occupancy sampling units should be surveyed at least three times if detection is less than 0.5 (MacKenzie and Royle 2005). Reasonable rates of detection, dependent upon species ecology, are required in order to have confidence in occupancy rates. Yet the number of surveys will also be dictated by the general levels of occupancy. If occupancy is high, more colonies of interest could be surveyed (relative to higher required sampling at lower occupancy) within a given field effort. MacKenzie et al. (2002) suggest that for two visits, occupancy should be greater than 0.7 and detection greater than 0.3. In addition to temporally replicated counts, distance sampling and double observer methods can be employed to estimate imperfect detection (Nichols et al. 2009).

The most common occupancy models are the Poisson and negative binomial models. The Poisson model is thought to be best for conditional detection. Conditional detection, where “variability in N [abundance] induces heterogeneity in site-specific detection probability” is essentially site-specific detection and is dependent upon two processes: presence and availability (Royle and Nichols 2003). Therefore, abundance can be determined with conditional detection when the distribution of detection probability is determined. This model is especially useful when covariates associated with abundance are not observable. The Poisson model assumes that animals are randomly distributed across the landscape and there is a constant mean. This differs from the negative binomial model which allows the density of animals to vary spatially (Royle 2004). The Poisson model is easily extended by “modifying the prior distribution to accommodate extra Poisson variation” (Royle 2004). Royle (2004) discusses that a “natural prior distribution is the negative binomial.” The negative binomial model can easily accommodate covariates that are spatially explicit (Kery 2008). In addition, an advantage of collecting covariates with this model is that time-varying, site-specific covariates can be used (Kery 2008). This differs from classical capture-mark-recapture studies where a time-varying, site-specific covariate can only be recorded when the animal is captured. With an occupancy study, the site is still visited and these covariates can provide information on why a species is not detected (Kery 2008).

Assumptions for both the Poisson and negative binomial models are that detections are independent and the population is closed. The estimable parameters are ‘ $\lambda$ ’= mean of distribution on abundance or the density of animals in the sampling unit and ‘ $r$ ’= binomial sampling probability that an individual is detected or animal-specific detection probability. The derived parameters are ‘ $\Psi$ ’=occupancy probability and ‘ $p$ ’=detection probability. To generate estimates of abundance, the total number of spatial replicates, ‘ $R$ ’, is multiplied by  $\lambda$ . An estimate of the standard error follows the similar logic, ‘ $R \times SE$ ’. This simple area expansion works for discrete units like wetlands or when the effective area of the sampling units is known. As a general guide, the number of sampling units required for this type of estimation depends upon the detection probability, but generally  $R=100$  (Royle and Nichols 2003).

The models used in the estimation process for both occupancy modeling and repeat counts are similar; the negative binomial and the Poisson. The main assumptions being population closure, species distributions relevant to the model, and independent detection. In addition, two further assumptions exist with regard to detection such that detection is similar over space/time or the primary sources of variation in spatial/temporal detection are identified and measured as covariates.

It can be difficult to distinguish between the two sampling strategies of occupancy modeling and repeat counts because occupancy modeling entails employing repeat surveys to measure variation. The critical difference is the statistic or unit of interest: repeat surveys in an occupancy framework gather presence-absence data, while repeat counts alone deal with the actual numbers or counts of species to estimate abundance. Repeat counts allow researchers to estimate the degree that counts will vary over a specific time period. Similar to occupancy modeling, the interpretation of estimates from repeat counts is dependent upon the intervals of time over which the counts are applied (Nichols et al. 2009). Counts must be conducted over a sufficient amount of time in which substantial movement is expected (usually at least 24 hours) yet that an animal's range overlapping the survey unit has not changed during the sampling period. Repeat counts allow researchers to determine the point at which additional surveys provide no additional gains in precision for counts of animals, a characteristic that is of considerable value when initiating new monitoring programs in areas without baseline or pilot data.

Royle et al. (2005) make the case for incorporating estimates of detection into interpretation of both occupancy and abundance, as rarely can non-detection be interpreted as absence. False absences, or presence of individuals/species when it is not detected, can negatively bias estimates because there are actually more present than detected/counted (MacKenzie et al. 2002). Detection probabilities can vary as a function of site characteristics, time, environmental variables or result from observer error, sampling technique mismatch, species non-response, temporary emigration, as well as simply not occurring there.

Nichols et al. (2009) consider various components of detection relative to coverage and detection probabilities. Coverage refers to an animal's presence within the sampling unit and is comprised of two distinct probabilities relating to range and movement. Probability of spatial sampling,  $P_s$ , is the probability that an animal's area of use overlaps the sampling unit. This probability is dependent upon how sampling units are selected, the size of those sampling units, and the size/shape of the animals' home range included within the sampling unit. Probability of presence,  $P_p$ , accounts for temporary emigrations ( $1 - P_p$ ) from an animal's home range. Detection probabilities, or the probability that an animal is detected given that it is available, are also comprised of two components relating to the sampling technique and observer.  $P_a$  is the probability that the animal is available to the sampling technique. For example, the songbird needs to sing in order to be detected in the point count.  $P_d$ , or the probability of detection given presence and availability, is dependent upon the conditions or skills that allow the animal to be detected, i.e., the observers need to be able to hear and identify songbirds. In addition, vocal characterization (how loud a bird sings or how shrilly), habitat structure, and distance from the observer can affect this probability. Detection is usually a function of several of these components acting simultaneously (e.g.,  $P_d$ ,  $P_a P_d$ , or  $P_p P_a P_d$ ). Various combinations of sampling methods can be used to elucidate these probabilities, which are important to define when making inferences about the target population.

Even after the kittiwake monitoring program is designed and implemented, it will be important to evaluate how well the design is performing relative to expectation and the overall program objectives. Power analysis is useful for determining the effectiveness and limits of long-term monitoring (Hatch 2003). When initiating a power analysis, the main issue to consider is the biologically significant level of change (magnitude of the effect). Different monitoring programs have different standards. For example, Hatch (2003) described that the Breeding Bird Survey strives to detect a decline of 50% over a 25-year period with a probability of 90% (Sauer 1993). The North American Waterbird Conservation Plan strives to detect a 50% decline in colonial waterbirds over a 50-year period (Kushlan et al. 2000). More regionally, the Alaska Seabird Management Plan developed by the U.S. Fish and Wildlife Service (2009), aims to detect a 20% change from year to year with 90% power. For seabirds, this has been viewed as conservative but important when considering the amount of time required to implement reactive efforts.

When the data and analyses are determined adequate to answer monitoring objectives, its incorporation into a long-term protocol is the culmination of the collection process. Carefully developed long-term monitoring programs will provide valuable information about biological diversity over space and time (Yoccoz et al. 2001) and can be used for scientific, management, or conservation purposes.

### **III. Recommendations**

#### **A. Summary of Literature Review and Preliminary Data Analysis**

For Pacific populations of kittiwakes, widespread reproductive failures that reflect oceanographic conditions are common. Yet the long-term population effects of their frequent inability to successfully breed are not well understood. Our literature review did not reveal nor refute widespread population decline, but declines may be expected, based on estimated recruitment rates. To gain a clear picture of how kittiwake populations fluctuate over time, consistency in monitoring their populations needs to be developed and maintained. Photographic sampling is commonly used for colonial breeding birds and can alleviate some of the common challenges to avian population monitoring. At present, the lack of discernible, long-term effects from colony breeding failures could reflect true population stability, but may be a result of the lack of continuous monitoring or a failure to select an appropriate biological unit.

Results from our preliminary data analysis (Appendix A) indicate that there is large annual variation in counts, though to what degree this represents breeding effort or population size remains unknown. We have learned that the photographic process (e.g., framing, settings) can affect counts of nests/birds and analysis and that interpretation of photographic data without considering these sources of bias is risky. High photographic quality will aid in accurately representing the size of breeding colonies. Additionally, a rigorous sampling design will assist in separating any actual population declines from sampling variation associated with the photographic process.

#### **B. Kittiwakes as a Subject for Long-term Monitoring.**

Kittiwakes are a strong candidate for long-term monitoring in GLBA. While monitoring populations for species-specific information is relatively straightforward, population monitoring with an ecosystem perspective is also valuable. Because reproduction closely reflects environmental conditions, particularly surrounding surface waters, kittiwake monitoring can provide a unique perspective into the ecosystem. However, making inferences to the marine environment, which usually requires data on kittiwake productivity, can be sampling intensive and complex. In contrast, kittiwake monitoring requires evaluation during the incubation phase of reproduction, the most stable phase of their life history. Consequently, the sampling effort made explicitly on behalf of kittiwakes is realistic for long-term monitoring.

Logistically, kittiwakes have monitoring advantages over other seabirds as they are colonial, highly visible, and tend to aggregate loosely (i.e., one bird per ledge or one bird deep) on open cliff faces. Aspects of kittiwake reproductive biology such as high nest attendance and colony fidelity combine with reasonable detection rates to lend themselves well to photographic sampling. In GLBA proper, limited field work is necessary to photograph nests and birds at colonies during the incubation period (Table APP B.2). The nesting cliffs are steep and plot boundaries (important for consistent monitoring) can be located with ease.

While adults and nests at breeding colonies lend themselves well to photographic monitoring, assessing productivity is much more difficult (Hooge et al. 1998). Beyond the difficulty in detecting chicks through photos, the remote location of several colonies within GLBA complicates estimation of hatch date and, thereby, informed predictions of the most suitable time

to measure productivity. Collectively, these considerations suggest that monitoring productivity may not be feasible.

While the life history of kittiwakes is generally conducive to long-term monitoring, there are several sampling challenges to be addressed. Substantial variation in counts can arise during the photographic and counting process. Poor weather and unsuitable light conditions can narrow the window for successful photographic surveys, while boat-based photography can be highly affected by current, tides, and wind. These factors not only inhibit photography but may also influence counts by reducing photo quality. Counts may also be affected by the differential ability of individual observers to identify nests or birds. Lastly, from the study design perspective, kittiwakes do not usually move between colonies; therefore, breeding season dynamics first reflect immediate surroundings. As a result, multiple colonies may need to be surveyed to reveal the spectrum of reproductive status within GLBA.

### **C. Biological Units of Interest and their Justification.**

Multiple authors suggest that continued low productivity will have severe implications on Pacific populations of kittiwakes (Hatch et al. 1993a, Hatch et al. 1993b); however, no available studies indicate whether or the extent to which regional populations have declined. Only one study has described declining kittiwake populations, as measured by reductions in the number of nests and adults from 1981 to 1991 (Hatch et al. 1993b). With an abundance of colony breeding failures across geographic area and time, key considerations relevant to design of long-term monitoring in GLBA center around whether population decline has occurred or whether declines are anticipated but not yet observed as well as the importance of the selected metric for assessing trend (e.g., number of birds, or number of nests).

When monitoring population size at breeding colonies, Hatch and Hatch (1988) note that inferring actual change from nest counts is risky due to variability in daily and annual rates of nest building and attendance. They suggest that counts of nests mirror breeding effort rather than adult population size. However, we also know that there is high daily variation in counts of adult birds, which could influence inferences similarly. So, which is best? Theoretically, if a population is truly declining there should be an eventual reduction in both the number of nests and birds over time that reflects recruitment and abundance, holding density dependent effects on breeding probability.

Though there is debate over the most appropriate biological unit to reflect population status, monitoring objectives and logistical feasibility will also guide selection of the unit. There are several potential units for use in trend detection of kittiwake populations in GLBA that are associated with breeding colonies. SEAN could consider the number of adult birds, the number of nests, the number of chicks hatched, or the number of chicks fledged. These units can first be framed through a census, a sample of permanent plots, or a sample of random plots. Secondly, they can be bound by the spatial scale of sampling such as GLBA proper or Margerie Glacier. An alternative to measuring these units would be to design a study to establish an index to abundance. For example, if a relationship between colony area and abundance were identified (i.e., if density of birds at a given colony is estimable and varies little), measuring the colony area would be sufficient to make inferences about trends in abundance. However, use of an index requires continual evaluation to ensure confidence in knowledge of the underlying relationship between the index (e.g., colony area) and the parameter of interest (e.g., colony population size).

Statistically, there are biological units with more flexibility and less variation associated with sampling. Given that it may not be necessary to monitor productivity for trend detection, we can reduce our list of potential units down to the number of birds or the number of nests. These more tractable biological units can be monitored across a greater window of time and will likely be subject to fewer detection errors. There tends to be less daily variability in the number of nests than the number of birds, but we suggest documenting both, the requisite data for which are readily obtainable from the same photographic samples. Similarly, the minimal time required to acquire images would allow documentation of the entire colony in a short time period, so we suggest delineating sample plots from the photographs after full inventory of the colony. Lastly, we suggest monitoring multiple breeding colonies as kittiwake ecology first reflects immediate surroundings.

A different sampling technique would be required if productivity information is desired. We suggest randomly sampling plots for direct field counts and employing a time-to-detection technique where repeat counts are not conducted. Further discussion on such approaches is discussed by Nichols et al. (2009).

Another important consideration when selecting and constraining sampling designs and biological units is the amount of processing resources it will require to sample and analyze the data. To process an entire set of photos for a large colony the size of Margerie Glacier (~5000 nests) requires substantial time – approximately four to six weeks. While a complete colony census would be possible, sampling would be more feasible from a data processing standpoint. Randomly selecting an annual sample of plots is more intensive than establishing permanent plots. However, interannual variation may be attributable to non-demographic factors such as colony movement or changes in nest-site fidelity. For example, the southernmost plot at the Margerie Glacier colony was occupied during the 1990s but no nests were found in 2011 (Appendix B). A random sample would avoid bias from changes in colony configuration. However, if colony configuration is stable over time (i.e., predominantly influenced by the physiography of the colony site), annual random sampling would likely increase sampling error relative to a permanent plot sampling methodology. These dynamics are unknown at this stage; therefore, we suggest a random sample of plots from multiple colonies to gain the most comprehensive picture of GLBA kittiwakes.

To ensure consistent and comparable images over time, detailed information on photographic methods (e.g., specific frames, number of photos, focal length, subject distances, camera settings), processing (e.g., colony composites, counting), and storage (e.g., databases) must be explicitly prescribed in Standard Operating Procedures.

Robust trend monitoring of kittiwake populations will require a sound sampling design, consistent technical methods, accounting for sources of sampling variability, and observation of multiple colonies over the long-term. The protocol will need to be periodically evaluated and updated with respect to advances in technology and analysis techniques. The protocol should also consider exporting the data to other entities (e.g., through the USGS North Pacific Pelagic Seabird Database or USFWS North Pacific Seabird Colony Database), to meaningfully contribute to the broader conservation community's ability to detect trends at larger geographic scales.

## **D. Suggestions for Future Analysis and Field Sampling**

To guide future efforts in monitoring protocol development, we suggest the following further analyses and field sampling.

1. Determine which photographic parameters have the greatest influence on detection. A comparison of historical and 2011 data could be used to discern the most critical element of the photographic process (e.g., angle, light, focus, coverage, or splitting, see Appendix A for description of these survey-specific covariates). This analysis would directly inform the field SOPs, improve field efficiency, and increase the quality of the raw data (i.e., images) that ultimately inform population analyses.
2. Compare how different biological units (e.g., nests or birds), sampling unit selection, and sample size affect trend detection and estimates of abundance. Further analysis of historical and 2011 data could be used to identify which sampling and analytical approach will provide the most accurate assessment of the parameter of interest. This work would also provide sample size recommendations for the most appropriate monitoring technique (e.g., repeat counts, occupancy sampling/modeling, or double-observer counts).
3. Determine if an index to abundance can be identified and should be implemented.
4. If measures of productivity are desired, evaluate the feasibility of monitoring at critical chick developmental stages. Specifically, evaluate how a time-to-detection technique performs at each developmental stage and compare results to chick detection using photographs. Determine whether photography may be used to evaluate productivity.



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## **Appendix A. Inventory of Historical Data and Preliminary Analysis**

### ***A. Scanning Slides***

We acquired slides from monitoring efforts in the 1990s at GLBA. All slides have been scanned at 2900 and 2400 dpi resolutions. We used CanoScan 9000F and Nikon CoolPix IV scanners. We tested scanning slides at multiple resolutions to determine which had the best clarity for viewing birds/nests. 2400 and 2900 dpi appeared to be the best while 4800 dpi applied a smoothing algorithm such as nearest neighbor interpolation which made small discrete objects too blurry. To test for sufficient resolution for identifying nests, we magnified the head of the kittiwake until it became pixilated. If the kittiwake head was composed of 20–30 pixels, the resolution was sufficient. In this, we consulted a professional photographer with experience examining historical photographs for minute changes. He suggested the resolution was adequate for this purpose and that this magnifying trick was a good way to confirm the resolution (K. Tape pers. comm.). The approximate size of each compressed .jpeg photo was 3–5 mb. The approximate size of each .tif photo was 69 mb.

We scanned approximately 1300 slides from 1993 to 1995. The scanning process took approximately one minute per slide. There were a number of missing slides, particularly from 1994, which have not been located at the time of this publication. Hooge et al. (1998) compiled a slide inventory, 'Slidetemp.xls' and 'Nests.dbf', to indicate which slides were used to count objects in each plot. The authors did not compile counts for individual photos but presented only cumulative plot counts. We used this inventory to organize photos by plot.

The original plots were established by Hank Lentfer (NPS), in 1990, along mineral extrusions, promontories, and crevices. There are 14 total plots and 10 subplots within. The authors used these plots to frame photos and partition counting then summed across the colony to achieve a census count, resulting in the entire colony as the unit of interest. Consequently, individual plots are not of biological interest nor are there enough plots for an adequate sample though the number of nests/birds far exceeds minimums necessary. To examine occupancy, change in colony configuration, or total area occupied, units would need to be generated that represent a biological area then the photos would have to be resampled accordingly; with respect again to monitoring objectives. Similarly, to examine the feasibility of and conduct estimates of the population or reproductive success, subplots and resampling are necessary.

### ***B. Processing Photos and Conducting Photo Counts:***

After organizing photos into plot folders, we stitched adjacent photos into a panorama using Adobe Photoshop. We checked for stitching errors and parallax by examining individual layer masks created in the panorama. After error checking, we brought them into a GIS in ArcMap 10.0 as a raster minus spatial information. We then compiled a plot composite with the greatest coverage at equivalent distances and/or focal lengths. This plot composite provided a frame of reference to map the photos associated with each count. To make sure that this composite accurately represented the plot, we compared it to the original plot boundaries developed by H. Lentfer in 1990, which we had scanned from the early monitoring protocol.

To subsample the colony and provide an area expansion, we set up smaller plots that were equivalent in size. We outlined the total area occupied by the colony and assigned subplots,

including at least one kittiwake nest in each subplot. To ensure that subplots were consistently sized among various plots, we selected photos at similar distances to delineate the subplots, then identified the physical features associated with the plots, and finally manually outlined these onto subsequent photos. This process cannot be automated due to the changing perspective of each individual photograph. The final map, 'Plot Boundaries.mxd', included individual plots and subplots and was used as a base map in mapping count photos.

To perform the actual counts, we selected those photos that best represented the plot and again stitched panoramas. We created multiple maps by plot and year, i.e. 'Plot 1 1993 Counting Kittiwakes.mxd', and inserted new dataframes for each count. A dataframe held all the photos by date to be used for counting resulting in five dataframes for each map that each held  $\geq$  five photos. We spatially aligned photos in each count according to the basemap, 'Plot Boundaries.mxd' and outlined areas of overlap to avoid double counting. Next, we outlined the subplots, created a plot point layer, and finally attributed the total points for each subplot.

### ***C. Data Analysis Methods and Results***

As of July 2011, we processed two plots in the Margerie Glacier colony, Plot 1 and Plot 6. Plot 1, along the very southern terminus of the colony, has 16 subplots while Plot 6, in the middle of the colony, has six subplots. To date, we have conducted photo counts of kittiwake nests by subplot, not yet of actual birds. There are a total of 21 sampling units, or sites, and five sampling occasions.

We assumed population closure for counts over the sampling period. In addition, we made the assumption that nest counts were not likely to vary by time of day. Therefore, for this initial analysis, we assumed that most of the variability was due to the photographic and counting process. To investigate sources of variability in detection, we collected covariate information on the photographic process. Our site-specific covariate, 'QUALITY', was an indicator of the overall quality of plot photographs and essentially an average of all survey-specific covariates. These survey-specific covariates included accounting for the completeness of the photographic coverage. Differences in coverage, 'COVERAGE', existed among years and were common; therefore, we made the assumption that there were no nests where there was no coverage. Other survey-specific covariates included lighting, focus, split counting, and perspective and were evaluated on each photo then summed by plot and corrected by number of photos. Unsuitable light conditions, 'LIGHT', led to issues with shadow and overexposure that might have inhibited counting. Bright sunlight simultaneously overexposed kittiwake heads and perch washes and created shadow where irregularities in the cliff face loomed over objects. Historical photos were usually out of focus, 'FOCUS', along the edges of the frame, which blurred objects together. At times, photographs overlapped or were split in areas, 'SPLIT', where birds/nests were densely aggregated. This made counting difficult because subplots were divided between photos and flipping back and forth could result in double counting or missing nests. Lastly, the position of the boat and photographer, 'ANGLE', interacted with the cliff face so that perspectives were highly variable. Certain portions of the cliff face, such as preferred nesting crevices, were not always visible in each count.

We used the Royle negative binomial count model (Royle et al. 2005) to estimate abundance, which uses repeat counts in an occupancy framework. The statistic is the actual count of birds/nests per unit area which estimates  $\lambda$ , the density of animals in the sampling unit. Detection

is estimated by ‘ $r$ ’, the binomial sampling probability that an individual is detected. The real parameters are  $r$  and  $\lambda$  while the derived are  $p$  (probability of detection) and  $\Psi$  (probability of occupancy). Assumptions for this model are population closure and that detections are independent and ‘ $p$ ’ is not time-dependent. To generate estimates of abundance ( $N$ ), we took the number of spatial replicates ( $R$ ) and multiplied by  $\lambda$  (Royle 2004). An estimate of the standard error follows the similar logic ‘ $R \cdot se$ ’. Accordingly, the sampling units are the equivalently sized subplots for this analysis. We used Program Mark (White and Burnham 1999) and Program Presence (Hines 2006) to conduct the analyses and formatted the input file to include or exclude certain covariates. We dealt with missing photos by indicating “.....” in Program Mark and “-----” in Program Presence. Group comparisons such as year to year variation are not presently allowed with this model (Royle et al. 2005). This restricted our analysis to individual years which reduced the sample size for models of constant counts.

We developed a candidate model set where counts varied by site-specific covariate in Program Mark and survey-specific covariates in Program Presence. We tested if the variation in counts was constant or was a function of the photographic process. There is currently no goodness-of-fit testing for the Royle negative binomial model though Kery et al. (2008) showed that it performed well with covariates in a Bayesian framework. We used Akaike’s Information Criterion (AIC) to select among competing models (Burnham and Anderson 2002).

Our analysis within Program Mark was unsuccessful in that the likelihood surface would not converge. We suspect this is because there is really high variability of nests within subplots and there was not enough data to categorize this variability. For example, some subplots had 0.0 nests per year while others had upwards of 50 nests per year. This range of counts would make the likelihood surface difficult to form in the estimation process. However, a similar analysis was possible in Program Presence. Within Program Presence the best approximating model for 1995 included the effects of the site-specific covariate ‘QUALITY’ (our collective assessment of photographic quality for that plot) on both detection and lambda (Table APP A.1). This model was the best approximating model in all three years, 1993–1995 (Tables APP A.1–3).

We present estimates from the best approximating model,  $\lambda = B_0 + B_1(c) + B_2(\text{QUALITY})$ , which was the top model in all three analyses by year. In 1995, detection probability ( $c$ ) was 0.53 while average abundance/sampling unit ( $\lambda$ ) was 8.97. For our derived parameters, occupancy ( $\Psi$ ) was 0.99 and our area expansion for abundance ( $N$ ) was 188.38. For 1994,  $c=0.75$  and  $\lambda=8.23$  while  $\Psi=0.99$  and  $N=172.74$ . During 1993,  $c=0.65$  and  $\lambda=8.13$  while  $\Psi=0.99$  and  $N=170.63$ .

Table APP A.1. Results from occupancy analysis of 1995 photo counts of Plot 1 and Plot 6, at black-legged kittiwake breeding colony, Margerie Glacier, Glacier Bay NP, Alaska.

Model	AIC <sup>a</sup>	$\Delta$ AIC <sup>b</sup>	AIC wgt <sup>c</sup>	$\mathcal{L}^d$	k <sup>e</sup>	2*LogLik
QUALITY c and $\lambda$	756.49	0	1	1	3	750.49
COVERAGE c	800.99	44.5	0	0	3	794.99
COVERAGE c and $\lambda$	800.99	44.5	0	0	3	794.99
QUALITY c	825.98	69.49	0	0	3	819.98
constant c and $\lambda$	903.77	147.28	0	0	2	899.77

Notation: ‘ $c$ ’=detection probability, ‘ $\lambda$ ’=density/sampling unit, ‘QUALITY’= site-specific covariate indicating overall quality of photographic parameters for plot, and ‘COVERAGE’, ‘SPLIT’, ‘FOCUS’, ‘ANGLE’, and ‘LIGHT’ = survey-specific covariates demonstrating quality of specific parameters of photographic



process. Column headings: 'a'=Akaike Information Criterion, 'b'= delta AIC, 'c'=AIC weight, 'd'=model likelihood, and 'e'=number of parameters.

Table APP A.2. Results from occupancy analysis of 1994 photo counts of Plot 1 and Plot 6, at black-legged kittiwake breeding colony, Margerie Glacier, Glacier Bay NP, Alaska.

<b>Model</b>	<b>AIC<sup>a</sup></b>	<b>ΔAIC<sup>b</sup></b>	<b>AIC wgt<sup>c</sup></b>	<b><math>\mathcal{L}^d</math></b>	<b>k<sup>e</sup></b>	<b>2*LogLik</b>
QUALITY c and $\lambda$	543.74	0	1.00	1.00	3	537.74
SPLIT c	555.68	11.94	0.00	0.00	3	549.68
FOCUS c	557.1	13.36	0.00	0.00	3	551.1
ANGLE c	559.02	15.28	0.00	0.00	3	553.02
LIGHT c	560.93	17.19	0.00	0.00	3	554.93
COVERAGE c	619.63	75.89	0.00	0.00	3	613.63
COVERAGE c and $\lambda$	619.63	75.89	0.00	0.00	3	613.63
constant c and $\lambda$	674.77	131.03	0.00	0.00	2	670.77
QUALITY c	810.22	266.48	0.00	0.00	3	804.22

Notation: 'c'=detection probability, ' $\lambda$ '=density/sampling unit, 'QUALITY'= site-specific covariate indicating overall quality of photographic parameters for plot, and 'COVERAGE', 'SPLIT', 'FOCUS', 'ANGLE', and 'LIGHT' = survey-specific covariates demonstrating quality of specific parameters of photographic process. Column headings: 'a'=Akaike Information Criterion, 'b'= delta AIC, 'c'=AIC weight, 'd'=model likelihood, and 'e'=number of parameters.

Table APP A.3. Results from occupancy analysis of 1993 photo counts of Plot 1 and Plot 6, at black-legged kittiwake breeding colony, Margerie Glacier, Glacier Bay NP, Alaska.

<b>Model</b>	<b>AIC<sup>a</sup></b>	<b>ΔAIC<sup>b</sup></b>	<b>AIC wgt<sup>c</sup></b>	<b><math>\mathcal{L}^d</math></b>	<b>k<sup>e</sup></b>	<b>2*LogLik</b>
QUALITY c and $\lambda$	530.98	0	1	1	3	524.98
COVERAGE c and $\lambda$	572.85	41.87	0	0	3	566.85
ANGLE c and $\lambda$	592.51	61.53	0	0	3	586.51
FOCUS c and $\lambda$	595.6	64.62	0	0	3	589.6
SPLIT c and $\lambda$	603.62	72.64	0	0	3	597.62
COVERAGE c	604.6	73.62	0	0	3	598.6
QUALITY p	613.37	82.39	0	0	3	607.37
constant c and $\lambda$	614.41	83.43	0	0	2	610.41

Notation: 'c'=detection probability, ' $\lambda$ '=density/sampling unit, 'QUALITY'= site-specific covariate indicating overall quality of photographic parameters for plot, and 'COVERAGE', 'SPLIT', 'FOCUS', 'ANGLE', and 'LIGHT' = survey-specific covariates demonstrating quality of specific parameters of photographic process. Column headings: 'a'=Akaike Information Criterion, 'b'= delta AIC, 'c'=AIC weight, 'd'=model likelihood, and 'e'=number of parameters.

During the analysis, the maximum likelihood estimation process would not tolerate some models. We excluded from the results table models with negative AIC values, a result of lack of convergence. The weight of evidence is not in favor of any models including individual survey-specific covariates (Tables APP A.1–3). The fact that these are not consistently present in the top model results (i.e.,  $AIC \leq 7.0$ ), but do appear occasionally suggests they are representing the interaction of possible spurious effects. The overall site-specific covariate, 'QUALITY', had the strongest bearing on the variability seen in repeat counts. The selection of this covariate, being a

function of all survey-specific covariates, in the best approximating model indicates that while individual aspects of the photographic process do not seem to alter estimates of  $\lambda$  and detection, collectively, they are critical.

Comparing the results of this analysis to summary statistics, we see that average counts per sampling units are 9.6, 12.3, and 10.1 from 1993 to 1995, respectively. An area expansion to abundance shows that  $N$  is 205, 259.8, and 213.5 from 1993 to 1995, respectively. This comparison demonstrates that not accounting for detection in the estimation process actually overestimates both the density of animals per sampling unit and the subsequent inference to abundance. Specifically applied to kittiwakes, by not accounting for variability in the photographic process we positively bias the population which could result in the perception that the breeding colony is larger than it actually is.



## Appendix B. Field Sampling 2011

Kittiwake sampling in GLBA took place in 2011 from June 20 to June 22. The field crew consisted of Chris Sergeant and Aleya Nelson. The objective of the field work was to test several methods of conducting photographic counts. From the analysis we have learned that most of the variation seen in the number of nests was a function of the photographic process. Photographic variables examined involved splitting counts between photos with high nest densities (SPLIT), lack of focus along frame edges (FOCUS), and coverage of photo plot (COVERAGE). Methodological variables included shadow resulting from direct sun exposure (LIGHT) and obscure cliff angles (ANGLE). Both types of variables could be reduced or mitigated by manual adjustments during the photographic process.

The plan was to perform three repeat counts at a primary site, Margerie Glacier, and confirm the presence of nesting birds at several other historical colonies. Other colonies within the park had been noted to be declining in 1995, at the conclusion of the first protocol development. Other colonies were photographed as time allowed.

We were able to photograph four colonies in GLBA along the main channel and into the west arm. We conducted three repeat photographic counts for Margerie Glacier, Gloomy Knob, and Lone Island (Table APP B.1). South Marble Island was photographed one time only (Table APP B.1).

Table APP B.1. Timing and dates of repeat photographic counts conducted at multiple kittiwake breeding colonies in Glacier Bay NP and Preserve, Alaska, 2011.

Colony	Count 1 (June 20, 2011)	Count 2 (June 21, 2011)	Count 3 (June 22, 2011)
Margerie Glacier	1515–1750 [2 h 35 m]	0945–1150 [2 h 5 m]	0920–1105 [1 h 45 m]
Gloomy Knob	1240–1300 [40 m]	1300–1320 [20 m]	1200–1210 [10 m]
Lone Island	1135–1210 [35 m]	1400–1410 [10 m]	1250–1300 [10 m]
South Marble Island	NA	NA	1330–1400 [30 m]

Margerie Glacier – The afternoon count at the east facing Margerie Glacier colony was probably too late in the day for the conditions. The weather was sunny and as the sun was disappearing to the west it was directly in the camera lens. We experimented with distances to the colony, which were primarily dictated by the currents. We worked from the southern end of the colony to the northern end. It seemed like there were substantially more nests in the colony than there were in the 1990s. In addition, some cliff facets containing nests were extremely high, upwards of 300–400 feet. The original plot photos taken by H. Lentfer did not include these highest areas of the colonies. Consequently, we further delineated subplots in the field to aid in taking photos and include these new areas. Plot 1 did not contain any nests, which varies from the 1990s. There was evidence of recent nesting with washes and nest marks but no evidence of current nesting at all. There was some vegetation encroachment on the south end of the colony. It is thought that as the glacier recedes the kittiwake colony migrates to keep in close association. This is probably attributed to both the vegetation succession in the older deglaciated areas and also proximity of prime foraging habitat directly adjacent to the tidewater glacier. As the glacier calves it stuns prey and strong currents upwell these forage fish. It may be worth examining how the geographic orientation of the colony and vegetation interact with glacial succession over time.

Eagles were present in the colony and caused multiple disturbances on the 1<sup>st</sup> and 3<sup>rd</sup> counts. It is possible that there are several eagles nesting within the perimeter of the kittiwake colony. These birds generally flushed the kittiwakes from their nests but only for a short period. Glaucous-winged gulls are also nesting—usually above the kittiwakes or in the more vegetated areas. No disturbances from glaucous-winged gulls were documented.

On June 21, the second count was overcast in the morning and was the best for photographing (as indicated by the earlier protocol, Hooze et al. 1998). The last plots started out overcast, but when the clouds dissipated, the morning sun was shining directly on the colony, creating glare and shadow. Due to the kittiwakes natural coloring it is easy to overexpose the birds and much definition was lost when the sun shone brightly.

We experimented with the aperture at two plots during the third count. Plot 6 and Plot 11, which have a moderate number of nests. Apertures were f13 and f8, with varying shuttering speed. For f13 the shutter speed was below 1/120, which given the focal length was 100 proved to be problematic. The aperture f8 was more consistent but similar to the automatic setting. For the purpose of counting kittiwakes and the high number of photos involved, using manual settings do not really provide a counting advantage and may really increase the amount of time photographing.

We visually estimated (use a range finder in the future) our distance to the colony to be between 150 and 300 feet. We used a 100–400x lens for the first pass at the colony but this may have been too high. It greatly increased the number of photos at the plot to get full coverage. We first photographed the plots at 100mm focal length then increased the focal length to 200 and 300 mm where nests were higher in density or nest sites were at high elevations (about four different tiers). The plan is to use the 100 mm photos to conduct the analysis and then validate with the 200–300 mm photos where needed. To get a larger perspective, we shot the colony at 50 mm (sometimes 135) from north to south on the way down bay. No birds were documented on Plot 14. For the analysis, we should compare 50–100 mm.

Gloomy Knob – The colony is located on the west side of the island, approximately halfway up the bay. Though the sun was directly on Gloomy Knob for the first count, we did not have too much glare due to the colony being at a lower elevation. We worked the colony from the south to the north for the first count, but from the north to the south for the two subsequent counts. The first count was photographed at 100 mm with a few 200 mm shots. The second and third were shot at 50 mm and 135 mm, which were good focal lengths for this colony. Distance was usually about 100 m from this colony. Lots of Brandt's cormorants and a few nesting glaucous-winged gulls above the colony and in the more heavily vegetated areas. A nesting bald eagle was located south of the colony, with the mate on the very south end of Gloomy Knob (same spot two days in row).

Lone Island – This colony was thought to be declining in 1995. There seemed to be a large number of nests for such a small island. An eagle flushed from the island when we approached for our first count (juvenile) but was not there for the subsequent counts. Apparently, there was an eagle nest here in 1995. Our first frames from count 1 were taken from the east side of the island. However, all of the nests were on the north side so for subsequent counts all frames were

from this perspective. We noticed two harbor seals and a pup on the island for the second and third repeat counts.

South Marble Island – This colony is the southernmost colony we surveyed. There were approximately 500 kittiwake nests for our subsequent survey. Concentrated nesting was on the east side of the colony with a few satellite pockets on the southern side including a small islet. There were also glaucous-winged gulls on nests.

For all colonies a total of two memory cards were required, an 8 gb which we filled supplemented by a 16 gb memory card. The total photo storage required was 9.23 gb. We transferred photos to a hard drive each evening of field sampling. Two copies of the photos went to SEAN NPS with C. Sergeant by both flash drive and the original camera memory cards.





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